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Title: Health Impact Assessment of Transport Policies in Rotterdam: Decrease of Total Traffic and Increase of Electric Car Use

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Abstract

Background: Green house gas (GHG) mitigation activities can be supported by showing co-benefits on health.

Method: Health Impact Assessment (HIA) was used to quantify co-benefits of local GHG mitigation policies in Rotterdam. The effects of a) 10 % reduction of private vehicle kilometers and b) 50 % electric-powered private vehicle kilometers, on particulate matter (PM_{2.5}), elemental carbon and noise were modeled and related health effects were assessed in Years of Life Lost (YLL) and Years Lived with Disability (YLD). The baseline was 2010 and the end of the assessment 2020.

Results: The exposure modeling indicates a reduction of PM_{2.5} by 40 % from 2010 to 2020 which results in 2,097 (CI (Confidence Interval) 1,403-2,711) YLLs for total mortality excluding accidents. EC was used as a sensitive indicator for air pollution near road traffic, which will be reduced by nearly 60 % in 2020. The policy aiming at reducing traffic is associated with a decrease of people exposed to noise which results in a prevention of 21 (CI: 11-129) YLDs due to annoyance and 35 (CI: 20-51) YLDs to sleep disturbance. The effects of 50 % electric-powered car use is slightly higher with 26 (CI: 13-116) and 41 (CI: 24-60) YLDs, respectively.

Discussion: Despite small co-benefits the results support the activities of the city Rotterdam to reduce GHG by taking actions in the transport sector. Innovative HIA estimates should include EC to consider road traffic intensity.

5 keywords: Air pollution; Disability-Adjusted Life Years; Health Impact Assessment; Noise; Transport

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1 Introduction

There have been many recent efforts to reduce greenhouse gas (GHG) emissions at national and international level following the creation of the United Nations Framework Convention on Climate Change (UNFCCC) in 1992, for example the 1997 Kyoto Protocol and the EU Directives (2003/87/EC, 2009/29/EC). Alongside energy production and the building sector, the transport sector is a leading contributor to carbon dioxide (CO₂) emissions. It represented around 22% of worldwide CO₂ emissions in 2010 (International Energy Agency, 2012).

Which plans have been developed to address the local impacts of GHG for cities? How can decisions on reduction of GHG emissions that are related to the transport sector be assessed at a local level? The answer to the first question leads us to consider the development of Health Impact Assessment (HIA) methodologies, whereas the second question directs us to consider the way to operationalize HIA in a real case and use the results for policy making. This article describes the path from addressing a general issue such as GHG in real scenario of planned interventions in a medium size European city to its effects on population health.

In the transport sector, decisions concerning measures to reduce GHG emissions can have both positive and negative environmental, social, economic and health effects (Haines et al., 2009; Thomas et al., 2014). Transport, and especially road transport enables access to employment and social life as well as essential services, like medical treatment (Thomson et al., 2008). However, modern transportation also brings some disadvantages from an increased need for vehicles and roads. An increase in vehicles leads in turn to an increase of exhausted fumes, and in order to build more roads, more

land and natural resources have to be used. Increased motorisation also increases the risk of road accidents. Therefore we propose that a systematic assessment of the impacts and especially health impacts is needed to inform the decision-making process. We propose HIA is a helpful tool to produce a comprehensive evaluation, with the aim of maximising health gains, of the consequences of decisions. HIA can help to advise policy makers and especially the non-health care related sectors by showing the effects of interventions on health (Kemmer, 2004; Mindell et al., 2003). In order to do so, we suggest that there is the need to integrate HIA of air pollution and noise exposure and related health effects with traffic modeling (Dora and Phillips, 2000; Negev et al., 2012).

Several recent studies have addressed the impacts of transport policies on health (e.g. Dhondt et al., 2013; Hosking et al., 2011; Schram-Bijkerk et al., 2009; Thomson et al., 2008). Transport increases air pollution levels, noise and the risk of accidents which in turn are responsible for several health outcomes, such as increased risk of cardiovascular or respiratory disease due to air pollution (e.g. Brunekreef and Holgate, 2002; Hoek et al., 2013; Künzli et al., 2000; Pope III and Dockery, 2006), injuries caused by dangerous driving behavior (Peden et al., 2004), traffic noise related mortality (Tobias et al., 2015) and annoyance caused by traffic noise (Miedema et al., 2011; Miedema and Oudshoorn, 2001). Yet, attributing health effects to transport policies is problematic due to data limitations, variations in methodologies, and equipment used for measuring pollutants such as air pollution and noise. In addition there is a need for detailed health statistics to calculate exposure-response functions, based on pooled and averaged estimates published in the scientific literature, which are then applied to local contexts.

Context and study population

The city of Rotterdam, located 20 km from the coast of the North Sea in the west of The Netherlands, has planned and implemented a series of measures to decrease CO₂ emissions by 50% between 1990 and 2020 as part of its GHG mitigation policies. These measures include more biomass burning in energy production, insulation of buildings to reduce the demand for energy and traffic related measures. The latter consists for a) 10% reduction of private vehicle kilometers on inner-urban roads and b) 50% electric-powered private vehicle kilometers on inner-urban streets by the year 2020 (Keuken et al., 2014). It is noted that this is not a realistic scenario for 2020 as currently less than 5 % of private vehicles kilometers driven are by electric vehicles. As the city of Rotterdam (and other cities) expected a considerable increase in electric road transport, this ambitious scenario was included to assess the impact on air quality, noise and health. These measures support the attempt by the city to become clean, green and economically robust by reducing noise levels and improving air quality with the aim of protecting the health of Rotterdam's population (Rotterdam Office for Sustainability and Climate Changes, 2011). Besides the two GHG mitigation policies an additional scenario was modeled: a business-as-usual (BAU) scenario. The BAU represents a real scenario that includes all transport-related measures that are already planned by the local authorities up to 2020. The BAU scenario includes the assumptions that the consequences of today's exposure and behaviour will continue without any changes to the year 2020, furthermore it includes regulations which are already decided but not yet implemented, like the exhaust emission standard Euro 6.

The aim of our study was to present an assessment of health co-benefits of GHG mitigation policies in the transport sector in Rotterdam. Effects of these policies were evaluated by comparing the burden of disease attributable to air pollution and traffic noise in 2010, chosen as a baseline, and the modeled burden when the policies will be implemented in 2020. Additionally the impact of the measures is compared to the BAU development.

2 Material and methods

For air quality, particulate matter ($PM_{2.5}$) has been identified as an indicator (WHO, 2006). There is compelling evidence on adverse health effects due to $PM_{2.5}$ exposure. (Brook et al., 2010; Hoek et al., 2013; WHO, 2006; WHO European Centre for Environment and Health, 2013). $PM_{2.5}$ consists of a mixture of primary (soot) and secondary particles. The latter are formed in the atmosphere from natural and anthropogenic gaseous emissions, such as ammonia (agriculture), sulphur dioxide (energy and industry) and nitrogen oxides (traffic and other combustion processes). Hence, $PM_{2.5}$ concentrations are only partly related to large-scale traffic emissions of nitrogen oxides, while primary emissions of soot particles from road traffic contribute very little to the mass of $PM_{2.5}$. Consequently, $PM_{2.5}$ is not a sensitive indicator for local traffic emissions (Keuken et al., 2012). For assessing the health risk of air pollution near road traffic, black carbon or elemental carbon (EC) can be used as an additional indicator alongside $PM_{2.5}$ (Janssen et al., 2011a; WHO European Centre for Environment and Health, 2013). This is particular relevant when assessing the health effects of air pollution related to transport measures. EC as a specific indicator for road traffic emissions can only be applied for assessing health effects for people living close

to road traffic, which in Rotterdam accounts for around 3.8% of the total population (13,946 people). Thus in the HIA for Rotterdam, both $PM_{2.5}$ and EC have been examined to assess the impact of the two local transport measures on health.

In addition to air pollution, road traffic also causes noise. Noise levels increase with higher traffic volumes and speed (Hosking et al., 2011) but also vary by road surface, surrounding vegetation and vehicle type. For traffic noise, a weighted average over 24 hours is performed, assigning higher weights to the evening and night periods than to the day period. This weighting scheme takes into account that sleep disturbance is an important aspect of noise-related health impacts. The weighted average noise levels are called “ L_{den} ” (day-evening-night) and “ L_{night} ” (only during the night). L_{den} is associated with annoyance and hence an indicator for psychological well-being, while L_{night} is related to sleep disturbance and cardiovascular effects (Miedema et al. 2011). In the HIA for Rotterdam, noise annoyance (people exposed to L_{den} over 55 dB(A)) and sleep disturbance due to noise (people exposed to L_{night} over 50 dB(A)) has been used to assess impacts of the two traffic measures.

Data sources

Air pollution and noise

Modeling the spatial distribution of air quality ($PM_{2.5}$, EC) and noise (L_{den} , L_{night}) requires various input data: the road and motorway network in Rotterdam, the related traffic (i.e. volume, fleet composition and speed), meteorology (atmospheric stability, boundary layer height, wind speed and direction), background concentrations of air quality and buildings near roads and motorways. The road network, population density and traffic data are available from the traffic department in the city of Rotterdam. Data are

available at a national level and are based on the European emissions inventory for road traffic “COPERT” (Ntziachristos and Samaras, 2009; Velders et al., 2012).

Risk coefficients and demographic data

Exposure response functions (ERF) for noise and concentration-response functions (CRF) for air pollution are needed to combine exposure data with health outcomes (table 1). In a literature search ERFs/CRFs were found and selected based on their accuracy (meta-analysis and recommendations of international organizations such as the European Commission or WHO were preferred) and applicability (international ERFs/CRFs were used because they are more robust than local ones). Another selection criterion is the availability of related mortality and morbidity statistics.

Table 1: Selected air quality and noise indicators and their exposure response function for several health outcomes

Health outcomes (specific population)	ICD-10	Risk coefficients	Source
Due to PM_{2.5}			
All-cause mortality excl. accidents (30 years and older)	A00-R99	1.062 (95% CI 1.040, 1.083) per 10 µg/m ³	Hoek et al. (2013)
Mortality due to ischemic heart disease (30 years and older)	I20-I25	1.152 (95% CI 1.111, 1.196) per 10 µg/m ³	Krewski et al. (2009)
Lung cancer related mortality (30 years and older)	C33	1.09 (95% CI 1.04, 1.14) per 10 µg/m ³	Hamra et al. (2014)
RADs, Restricted activity days (18-64 years)		4.75% (95% CI 4.17, 5.33) per 10 µg/m ³	Hurley et al. (2005)
Due to EC			
All-cause mortality excl. accidents (30 years and older)	A00-R99	1.06 (95% CI 1.04, 1.09) EC per 1 µg/m ³	Janssen et al. (2011a)
Due to Noise			
Annoyance	-	%HA = $9.868 \cdot 10^{-4} (L_{den-42})^3 - 1.436 \cdot 10^{-2} (L_{den-42})^2 + 0.5118 (L_{den-42})$	Miedema and Oudshoorn (2001)
Sleep disturbance	G47	%HSD = $20.8 - 1.05 \cdot L_{night} + 0.01486 \cdot L_{night}^2$	Miedema and Oudshoorn (2003)

Noise: Percentage and number of adults highly annoyed, indoor, in 1 year. Percentage and number of adults highly sleep disturbed, indoor, in 1 year. CI=Confidence Interval

Besides the ERFs and CRFs, demographic, mortality and prevalence data are needed. Demographic data for the population of Rotterdam, stratified by sex and age (five-year age groups) for the year 2010, were obtained from the Gemeente Rotterdam Centrum voor Onderzoek en Statistiek webpage (2014) as well as all-cause mortality data on the same level of detail. Mortality and prevalence data were not available at a local level. Therefore national prevalence rates were used by assuming that these rates are similar to local rates. The data were available online from Statline (2014) and IKNL (2014). All-cause mortality related to long-term exposure to $PM_{2.5}$ and EC was calculated for people aged 30 years and older. Furthermore lung cancer and ischemic heart disease related mortality caused by $PM_{2.5}$ were estimated. A causal relationship is also available for noise and the health outcomes annoyance (Miedema and Oudshoorn, 2001) and sleep disturbance (Miedema et al., 2003).

To facilitate a comprehensive assessment, not only health outcomes are included. In addition, restriction in activity because of ill-health was gathered in terms of Restricted Activity Days (RAD). A RAD is a day when a person is forced to alter his or her normal activity for health related reasons. This very general health outcome includes days of work loss, bed disability and minor restrictions. These days are highly related to $PM_{2.5}$ exposure (Hurley et al., 2005; Ostro, 1987).

Key summary measures and statistical analyses

Environmental measures

The total road traffic volume in 2010 was 13 million vehicle kilometers per 24 hours. The average distribution of personal cars, light duty and heavy duty trucks is, respectively 90%, 5% and 5%. The percentages of the people living near inner-urban roads with

over 10,000 vehicles per 24-hr, within 100 m from motorways and the remainder of the population living at the urban background was respectively, 4%, 1% and 95%. The expected volume growth of road traffic by Rotterdam authorities to 2020 is zero on inner-urban roads and 2% per year on urban motorways. For dispersion of air pollutants at inner-urban roads in Rotterdam, a “street canyon” model was applied and for motorways a “line-source” model (Beelen et al., 2010; Keuken et al., 2012). In addition to the aforementioned data input, the dispersion models also require emission factors for road traffic in 2010 and 2020 for primary $PM_{2.5}$ and EC. Secondary $PM_{2.5}$ is also included by NO_x and NO_2 formation which is located outside of the city and thus is part of the background concentration of $PM_{2.5}$.

The contribution of local road traffic emissions is added to the background air quality concentrations and presented within a Geographical Information System (GIS) with the road network of the city. The annual average background concentrations of primary and secondary $PM_{2.5}$ and EC in 2010 and 2020 at a spatial resolution of 1*1 km are available in the Netherlands as part of the regulatory framework to assess the air quality in the Netherlands (Velders et al., 2012). This framework includes a calculation with a regional dispersion model which computes the contribution of large-scale emissions as provided in the European emission inventory database to annual average air quality in the Netherlands. On top of the European contribution, a national dispersion model is applied in combination with the national emission database to add the national source contributions to the background concentrations in the Netherlands. It should be noted that the emissions from local road traffic are already included in the background concentrations in each grid cell of 1*1 km. Consequently, adding local traffic emissions

in a specific grid cell results in “double counting” the road traffic contribution to air quality. However, the contributing to the background concentrations are equally distributed over a grid cell of 1*1 km, while the contribution to air quality near the road network in each grid cell is computed by a street canyon model for inner-urban roads and a line-source model for motorways (Beelen et al., 2010; Keuken et al. 2012). For inner-urban roads, the double counting is negligible as the traffic intensity is relatively limited as compared to the size of a grid cell of 1*1 km, but for motorways with relatively high traffic intensity the air quality is corrected for double counting (Velders et al., 2012).

Noise calculations in Rotterdam have been carried out in two steps: firstly, calculating the emission and then the transmission. The emission calculations take into account traffic and road characteristics: traffic intensity, traffic composition (percentages light duty, medium duty, and heavy duty vehicles), speed, road height, and road surface. Similar to air quality, emission factors have been established in the Netherlands for road traffic and distinguishing emissions from tyre and engine noise. The transmission calculations take into account the distance between source (road) and building facade, air and ground attenuation, annual average meteorological conditions and reflection of objects opposite the building. These calculations result in the noise exposure at the center of a building represented in decibels (dB). The noise exposure in Rotterdam was calculated with the Dutch standard method (SRM2). SRM2 is in accordance with requirements of the EU Environmental Noise Directive (European Parliament and Council of the European Union, 2002). Therefore, for $PM_{2.5}$, EC and L_{den} the exposure at residential addresses of the population was distinguished in people living in street

canyons with more than 10,000 vehicles per 24hrs, people living within 100 m from a motorway and the remainder of the population.

Summary measure of population health

The health impact of the policies has been expressed in Disability-Adjusted Life Years (DALY) to enable a comparison of the health effects (Schram-Bijkerk et al., 2009). This summary measure of population health includes both mortality (Years of life lost, YLL) and morbidity (Years lived with disability, YLD) in one measurement (Knol et al., 2009; Murray et al., 2012). One DALYs equals one lost year of health life. We used population data of 2010 for all calculations to enable a comparison of the burden of disease in 2010 and in the future scenarios. For each health outcome the impacts attributable to the exposure was calculated stratified by 5 years age groups and sex.

For long-term projection discounting of time is suggested (Remais et al., 2014), but in our assessment the prevented life years in the future have the same weight as the ones today. Likewise no age-weighting was applied.

First, the difference in the pollution concentrations between the measured air pollution concentration in the baseline scenario and the modeled concentration in the future scenario was calculated. Based on the CRFs the proportion of population risk which is attributable to these PM_{2.5} and EC concentrations was calculated (Prüss-Üstün et al., 2003). In the last step the death cases of the disease due to the exposure are multiplied by the life expectancy to obtain YLLs. The CRFs used were available together with 95% confidence intervals (CI) which were used to calculate an upper and lower bound for the results as part of a sensitivity analysis.

For noise the formulas presented in table 1 were used to calculate the number of people annoyed and sleep disturbed. These numbers were then multiplied by a Disability Weight (DW) to obtain YLDs. The DWs from the Environmental Burden of Disease study on noise of the WHO and the recommendations of the European Commission were used: for annoyance 0.02 (lower bound 0.01 and upper bound 0.12 (Miedema et al., 2011)) and for sleep disturbance 0.7 (lower bound 0.04 and upper bound 0.1 (Janssen et al., 2011b)). The lower and upper bound was used to calculate the CI.

The possibly synergistic effects due to interactions of PM_{2.5} and noise were not considered in the present HIA, despite some preliminary indications that such interactions exists (Schram-Bijkerk et al., 2009). Given that no studies have quantified these potential interactions we could not use them in this assessment (Babisch et al., 2014) . For detailed description of the calculation methods see appendix.

3 Results

Exposure data

As expected, people living near heavy road traffic are more affected by transport policies than the rest of the population. The annual average concentrations for PM_{2.5}, EC, L_{den} and L_{night} in 2010 are shown in Figure 1 and in Table 2 and 3.

Figure uploaded separately.

Figure 1: Modeled exposure data for air quality and noise in Rotterdam, 2010

Figure 1a illustrates that PM_{2.5} levels in the Netherlands and the rest of Europe (EEA, 2012) are dominated by the regional background and there is limited contribution of emissions from road traffic near motorways around Rotterdam and along inner-urban roads in the center of Rotterdam. Figure 1b illustrates that EC is a more sensitive

indicator for road traffic emissions than PM_{2.5}, as EC concentrations near intense road traffic are a factor of 2 to 3 times higher compared to the urban background in Rotterdam. Figures 1c and 1d illustrate that noise emissions from road traffic results in large parts of the population being exposed to L_{den} levels over 55 dB in Rotterdam.

Compared to the baseline model, the BAU 2020 scenario will result in substantially lower air pollution volumes measured both as PM_{2.5} and EC (Table 2). The estimate for PM_{2.5} in the baseline model was 15.2 and shifted downwards, a 40% reduction, to 9.1 under the BAU 2020 scenario. A larger reduction of 58% can be seen when considering the EC estimates; 1.2 at baseline and 0.5 under the BAU 2020 model. The effects of the two policies are marginal compared to the BAU scenario. For the total city no differences are apparent, only in the immediate streets are small differences notice by around 2% for PM and 25% for EC when comparing scenario b with the BAU scenario.

Table 2: PM_{2.5} and EC (µg/m³) in different scenarios and distributed in locations in Rotterdam

Scenarios	Location	background		motorway		street		urban		total city ^a	
		PM _{2.5}	EC	PM _{2.5}	EC	PM _{2.5}	EC	PM _{2.5}	EC	PM _{2.5}	EC
Baseline 2010		14.9	1.1	15.5	1.4	15.8	1.5	15.2	1.2	15.2	1.2
BAU 2020		8.9	0.5	9.2	0.5	9.4	0.8	9.1	0.5	9.1	0.5
Scenario a) 10% less traffic		8.9	0.5	9.2	0.5	9.2	0.7	9.1	0.5	9.1	0.5
Scenario b) 50% electric cars		8.9	0.4	9.2	0.5	9.2	0.6	9.1	0.5	9.1	0.5

^a population weighted average

In 2010 around half of the population in Rotterdam has been exposed to noise above 49.5 decibels and at night around a sixth is exposed to this level of noise. The number of people exposed to noise will probably increase in the future. In the BAU scenario more than 10,000 people more will be exposed to noise levels over 49.5 decibels. The increase is weaker in scenario a) and b), but still in scenario a) around 1% more people will be exposed to noise levels above 49.5 decibels compared to the baseline scenario.

Table 3: Noise exposed population (in %) grouped by exposure level and scenarios in Rotterdam

L_{den} / L_{night} in decibels	Baseline 2010		BAU 2020		Scenario a) 10% less traffic		Scenario b) 50 % electric cars	
	L _{den}	L _{night}	L _{den}	L _{night}	L _{den}	L _{night}	L _{den}	L _{night}
>=49.5 - <54.5	38.8	64.5	38.2	62.4	38.7	64.5	39.0	64.2
>=54.5 - <59.5	30.5	29.1	30.2	30.6	30.6	29.1	30.6	29.2
>=59.5 - <64.5	21.0	5.9	21.0	6.4	20.9	5.9	20.7	6.0
>=64.5 - <69.5	8.3	0.5	9.0	0.6	8.4	0.5	8.2	0.5
>=69.5 - <74.5	1.3	0.1	1.5	0.1	1.4	0.1	1.3	0.1
>=74.5	0.1	-	0.1	-	0.1	-	0.1	-
Total	100	100	100	100	100	100	100	100
Number of people	312,214	109,085	323,356	116,272	315,788	110,909	314,003	109,562

Comparison of the baseline scenario and the two policy scenarios

Based on the changes in the exposure to PM_{2.5}, EC and the increase of people exposed to noise, the related effects on health were quantified (table 4)

Table 4: Comparison of baseline scenario in 2010 with scenarios a) and b) in 2020

Health outcomes (specific population)	Affected population		Impacts		Metric s
	Scenario a	Scenario b	Scenario a	Scenario b	
Due to PM_{2.5}					
All-cause mortality excl. accidents (30+)	196 deaths	196 deaths	2,097 (1,403-2,711)	2,097 (1,403-2,711)	YLL
Mortality due to ischemic heart disease (30+)	37 deaths	37 deaths	346 (269-423)	346 (269-423)	YLL
Mortality due to lung cancer (30+)	27 deaths	27 deaths	282 (138-410)	282 (138-410)	YLL
RADs, Restricted activity days (18-65 years)	389,716 people	389,716 people	13,962 (11,976- 15,316)	13,962 (11,976-15,316)	RAD
Due to EC					
All-cause mortality excl. accidents (30+)	13,946 people	13,946 people	67 (46-98)	67 (46-98)	YLL
Due to noise					
Annoyance	- 352 people	- 85 people	-7 (-4- -42)	-2 (-1- -10)	YLD
Sleep disturbance	-141 people	-49 people	-10 (-6- -15)	-4 (-2- -6)	YLD

The health effects related to air pollution are 2,097 (CI: 1,403, 2,711) YLLs for total mortality excluding accidents which could be saved if 10% fewer vehicles would drive in 2020. Overall, per 1,000 adults 5.8 YLLs can be prevented. The results for mortality due to ischemic heart disease (346 YLLs, CI: 269, 423) and lung cancer mortality (282 YLLs, CI: 138, 410) will be smaller but still not negligible. Furthermore, by implementing the policy, 13,962 RADs (CI: 11,976-15,316) can be prevented. Comparing the baseline scenario 2010 with the policy scenario a) results in 0.7 µg/m³ less EC which could prevent 67 (CI: 46, 98) YLLs, equivalent to 7.0 YLLs per 1,000 people. The results for scenario b) are equal, because the two interventions have the same effect on PM_{2.5} and EC emissions.

Regarding traffic noise, a comparison of the baseline with scenario a) results in a loss of 7 (-4- -42) YLDs because of noise annoyance and a loss of 10 (-6- -15) YLDs due to noise induced sleep disturbance. Similar results are shown for scenario b) where 2 (-1- -10) YLDs will be lost due to noise annoyance and 4 (-2- -6) due to sleep disturbance caused by noise.

Comparison of the BAU scenario 2020 and the two policy scenarios 2020

To assess only the effects of the policies, they were compared to the BAU scenario. Effects of other transport policy developments were therefore not considered. Comparing the BAU scenario and policy a) no difference in the preventable burden of PM_{2.5} and EC, is visible. The same results show the comparison with scenario b).

Table 5: Comparison of BAU scenario in 2020 with scenarios a) and b) in 2020

Health outcomes (specific population)	Affected population		Burden of disease		Burden of disease unit
	Scenario a	Scenario b	Scenario a	Scenario b	
Due to PM _{2.5}					
All-cause mortality excl. accidents (30+)	0 deaths	0 deaths	0	0	YLL
Mortality ischemic heart disease (30+)	0 deaths	0 deaths	0	0	YLL
Mortality due to lung cancer (30+)	0 deaths	0 deaths	0	0	YLL
RADs, Restricted activity days (18-65 years)	0 people	0 people	0	0	RAD
Due to EC					
All-cause mortality excl. accidents (30+)	0 deaths	0 deaths	0	0	YLL
Due to noise					
Annoyance	1,073 people	1,340 people	21 (11-129)	26 (13-161)	YLD
Sleep disturbance	505 people	595 people	35 (20-51)	41 (24-60)	YLD

Positive effects can be attributed to the policies we investigated, if noise exposure is considered. Comparing the BAU scenario with scenario a) results in a preventable burden of 21 (11-129) YLDs due to less noise annoyance and 35 (20-51) YLDs due to reduced noise induced sleep disturbance. The effects of 50% electric car use are slightly higher. For annoyance 26 (13-161) YLDs and for sleep disturbance 41 (13-161) YLDs could be prevented by the increase of electric car use in the year 2020. Comparing the two interventions regarding noise, intervention b), 50 % electric cars in 2020, has a greater impact: a reduction of 5 more life years for annoyance and 6 more life years for sleep disturbance could be prevented.

4 Discussion

The aim of our study was to present the assessment of health co-benefits of two GHG mitigation policies in the transport sector in Rotterdam. First, the effects of the two policies on PM_{2.5}, EC and noise were modeled and then the related burden of disease was quantified in YLLs, YLDs and RADs. The results show demonstrable health impact but should be interpreted with caution due to several uncertainties which are described in the following sections.

Exposure data

The impact of local measures is limited as these measures are restricted to inner-urban roads which represent about 65% of the total traffic volume in Rotterdam and the remaining 35% of motorway traffic is not affected by local measures. The aim of the city of Rotterdam is to increase urban density in the city center (Schaminée et al., 2012), which is associated with an increase in the number of cars on the motorway around the city. The increase is estimated at around 2% per year for motorway traffic and zero for inner-urban roads. Therefore the intervention with the aim of decreasing the use of cars would nearly offset this change. It can be concluded that the policy will have impacts but due to other changes these impacts are rather small.

Local measures do not contribute substantially to PM_{2.5} reductions near heavy road traffic as direct traffic emissions hardly contribute to PM_{2.5}. For EC, apart from emission standards such as Euro-6, the introduction of electric vehicles is the most effective measure to reduce EC levels near busy inner-urban roads. However, the impact of the population-weighted average of this improved air quality near inner-urban roads is limited due to the low number of people living close to major roads.

The different components of PM_{2.5} were not considered in this assessment. However, they can have different effects on health (Hurley et al., 2005). Thus more information is needed to assess these effects as well as their combined effects to avoid double counting. A single-pollutant model is preferred, because the effects of PM_{2.5} seems to be robust to adjustment for other pollutants (WHO European Centre for Environment and Health, 2013).

EC has been used as a sensitive indicator for exhaust emissions from road traffic: in Europe emissions from diesel traffic contribute 70% of EC concentrations (Bond et al., 2013). In 2020, the levels of EC are expected to decrease due to Euro-6 which is the emissions limit for motor vehicles in Europe since 2014. Due to this BAU development, which will reduce EC levels substantially, the two additional CO₂ reduction measures beyond BAU taken in Rotterdam hardly contribute to further decrease EC levels. It is noted that these conclusions are based on modelling with relative large uncertainties in the input parameters: the traffic volume, the fleet composition and the emission factors for road traffic.

The exposure to noise is expected to increase due to the growth aim of the city. However, the policies will result in less people being exposed to noise levels above 50 decibels. A reason for the relative small impact of around 1% less people being exposed to this level of noise in scenario b) is that electric vehicles have no engine noise but still the tyres result in noise emissions.

Comparison of the baseline scenario and the two policy scenarios

Several life years (2,097 for all cause mortality and PM_{2.5}, 67 for all cause mortality and EC) and RADs (13,962) can be prevented by a decrease of PM_{2.5} and lower EC levels,

which shows that a decrease in air pollution can produce important health benefits and that Rotterdam is proceeding in improving air quality.

The comparison of the baseline scenario with the future scenarios gives negative results for the impacts of noise, which means that more healthy life years may be lost due to increasing noise exposure from 2010 to 2020. A reason for the increase of noise levels could be the general increase of vehicles in Rotterdam

Comparison of the BAU scenario and the two policy scenarios

When comparing the burden of disease in the BAU scenario with the burden of disease in the future scenarios, no effects related to air pollution and only minor effects due to noise (between 21 and 41 YLDs) can be predicted. An explanation for these results is to be assigned to the already implemented environmental standards in the city and an expected continued reduction of gaseous precursors of secondary PM_{2.5}, such as Nitrogen oxides and Sulfur dioxide (e.g. energy production, industry, refineries) and Ammonia (e.g. agriculture), whereas for EC road traffic emissions are expected to be reduced by the introduction of Euro-6.

The quantitative effects are rather small (between 21 and 41 YLDs for health effects of noise exposure), which is in line with other local HIA studies (Joffe and Mindell, 2002; Schram-Bijkerk et al., 2009). A direct comparison with other Dutch Environmental Burden of Disease studies, like Hänninen et al. (2014) or HIA studies would be inappropriate, because the results depend highly on the considered policies. Hence a comparison with similar policies and the same reference area would be meaningful, but no such studies are available.

The assessed policies were determined by the city itself and the results are therefore policy relevant (Giles et al., 2011). However, the policies are ambitious and achieving the goal of 50% electric vehicle kilometers driven seems unlikely though, in the light of a current share of electric cars by around 5%.

Strength and limitations

An uncertainty analysis is presented in the following section to give information on the reliability and consistency of the results. The guidance of the *International Programme on Chemical Safety* (IPCS, 2008) and from Remais et al. (2014) were followed by calculating CI of the quantifications and describing the uncertainties qualitatively.

Population data were gathered from representative national Dutch sources and thus sample uncertainties are probably very small. National morbidity and mortality rates were applied to the population of Rotterdam. It is assumed these rates can be applied to the population of Rotterdam. An aspect which needs to be considered concerning the CRFs of PM_{2.5} is that they are only applicable for a population which is exposed over a long period of time. But, there is a high percentage of the population of Rotterdam that is mobile, and foreign, and probably not exposed to Rotterdam's level of PM_{2.5} for their whole lifetime (Gemeente Rotterdam, Centrum voor Onderzoek en Statistiek 2013). This aspect could not be considered in our assessment, because only the general number of people having foreign roots was available and not the duration of their stay in Rotterdam.

ERFs and CRFs are crucial for the risk assessment, because they link the exposure to environmental factors with health outcomes. Thus, only health effects for which a statistical relationship between the risk factor and the health outcome were available

could be included. The assessment is therefore very conservative by including only the burden of a few health outcomes, which are already well understood. Therefore, if more data for the statistical relationships between risk factors and health outcomes and related health statistics like prevalence data were available more health outcomes could be included.

Not only the availability of ERFs/CRFs limits the assessment, but also the ERFs/CRFs themselves. The epidemiological studies, which derived them, restricted the assessment by including certain uncertainties (Knol et al., 2009). By linking exposure and health data with the help of ERFs/CRFs a mismatch of data can occur, because of different aggregation levels of the data. The health outcomes were assessed at individual level including individual cumulative exposures and on the ERFs and CRFs base on aggregate measures of pollution (HEIMTSA, 2010). The standard approach was applied by using CRFs based on outdoor exposure, because in this assessment outdoor exposure data were modeled. For an accurate match of health data and ERFs/CRFs ICD-10 was used. It was assumed that international CRFs are applicable for the population and situation in Rotterdam, because the composition of the pollutants and the exposure situation were similar. No local ERFs/CRFs were preferred, because ERFs/CRFs for long-term effects from international studies are usually more accurate than those from local ones. Furthermore, studies found a high consistency of international epidemiological results, even that they refer to different populations, climates and pollution mixtures (Hurley et al., 2005).

The results are based on a comparison of the baseline scenario and the policies as well as a comparison of the BAU scenario and the policies. The years in-between were not

considered in detail. Likewise, population changes were not considered, although it could be assumed that there will be an increase in the number of people in Rotterdam. Due to population projections the population will increase up to 640,215 people in 2020 (Hoppesteyn, 2012). The expected growth is due to internal and international migration as well as natural growth.

We calculated the overall burden attributable to PM_{2.5}, EC and noise for the population of Rotterdam. However exposure to air pollution and noise is not equally distributed, neither spatially nor socially. Evidence shows that the proximity to traffic-congested roads is related to higher environmental hazards (Kohlhuber et al., 2012). This leads to a decline of house prices nearer to busy roads. This in turn leads to segregation, because people who can afford it move away and only people who do not have such financial options stay. Consequently, people with lower socioeconomic status have a higher burden due to traffic (Bunge, 2008; WHO Regional Office for Europe, 2012). Thus, interventions concerning the reduction of environmental risk factors in the sense of environmental justice would improve the health of people who need it most. Additionally, such preventive interventions yield value for society as a whole in contrast to curative activities, because curative activities are used more by people who already have easier access to health care. By minimising environmental risk factors the health of all people can be improved and especially the most vulnerable groups would benefit the most (WHO Regional Office for Europe, 2012). Thus analyses for these groups as well as for children and elderly people, cyclists and pedestrians would be appropriate, which was not possible due to data limitations (Schaminée et al., 2012).

5 Conclusions

This study assessed the effects of GHG mitigation policies in Rotterdam by quantifying the impacts on health that might be expected from taking specific transport measures.

The local traffic measures to reduce CO₂ emissions from road traffic concern road traffic on inner-urban roads but *not* on motorways, as the latter are regulated by national authorities. Inner-urban road traffic represents about 65% of the total vehicle kilometers driven within Rotterdam. Hence the city of Rotterdam only has partial control over reducing CO₂ emissions from road traffic within its boundaries and thus consequently the impact on air quality and related health effects. In Rotterdam (and the rest of Europe), it is expected that the regional background of air pollutants in 2020 will be lower than in 2010 due to further application of cleaner technology. The air quality development of the city of Rotterdam is progressing well, but benefits of emission reduction measures may be partially offset by the increase of vehicle kilometers. Therefore it is worthwhile to consider, besides the regulatory measures community and individual interventions.

It is assumed that the noise levels in Rotterdam will increase due to 2% annual increase of circulation of cars each year. Thus noise exposure will remain an important aspect to be considered for Rotterdam. Additional interventions to minimise noise could be considered, such as silent asphalt for roads.

While the estimated impacts on health are rather small, they can nonetheless be used to inform policy makers on the co-benefits of the two policies which were developed with the primary aim of reducing GHG emissions. The policies were discussed by the city of

Rotterdam and are therefore policy relevant and the results can be readily applied in the decision making process.

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